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Successful reintroductions of the endangered long-lived Sargent's cherry palm, *Pseudophoenix sargentii*, in the Florida Keys

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ABSTRACT

The 1991–1994 reintroductions of Florida endangered *Pseudophoenix sargentii* to 13 Florida Keys sites represent a rare example of a successful multi-agency long-term effort to conserve a long-lived palm. To assess reintroduction success, we compared population demographics with and without reintroduced plants and conducted population viability analyses. Since 1991, the wild population has increased 6.4-fold. Survival from 2000–2004 was 94%, growth was positive ($\lambda = 1.013$), and there was no predicted extinction risk. Recent wild population growth is attributed to good seedling recruitment and removing the greatest threats. After 14 years, reintroductions had 43% survival, increased total plants in the wild by 27%, and expanded the species' distribution. Reintroduced plants had faster maturation rates, improved population age structure, and enhanced population growth ($\lambda = 1.032$). Success varied with transplant year, location, microsite, and original transplant size. Failures in 1991 and at some historic sites emphasize the need for a multi-year, multi-site approach to reintroductions to buffer against stochastic losses. Rockland hammocks and the tops of coastal berms had greatest plant growth and survival. Large transplants had the greatest survival. Because no reintroduced plants are reproductive, transitions between stages are extremely slow, and plants may require >30 years to mature, continued institutional dedication to long-term monitoring will be required to assess whether the populations are self-sustaining. Horticultural expertise and ex situ collections complimented support of land managing agencies for the species' preservation. These first rare plant reintroductions to Florida State Parks opened avenues for more plant conservation efforts and public interpretation.

1. Introduction

Reintroduction has been encouraged as an extinction prevention strategy for plant species (Maunder, 1992; Falk et al., 1996; ANPC, 1998; IUCN, 1998; Clewell et al., 2000), however, examples of effective reintroductions that result in self-sustaining populations are limited (Griffith et al., 1989; Falk et al., 1996, but see; Rich et al., 1999). Prior to reintroductions, knowledge of the species' taxonomy, reproductive biology, demography, horticulture, and ecology are advised (Griffith et al., 1989; Falk et al., 1996; ANPC, 1998; IUCN, 1998; Clewell et al., 2000; SER, 2002) and lack of understanding of these factors have

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accounted for many restoration failures. As practitioners have begun to understand these pre-restoration factors, the success of restorations has improved. However, experience has shown that many years are required to evaluate the success of reintroductions, especially for long-lived species (Maunder, 1992; Schwartz, 2003; Maschinski, 2006).

For species on the brink of extinction, there is an urgent need for conservation, and reintroduction may be a vital component. Such was the case of *Pseudophoenix sargentii*, the Sargent's cherry palm (*Arecaceae*), which was endangered due to habitat destruction, land conversion to agriculture, grazing, over-harvesting, and hurricanes (Ledin et al., 1959; Johnson, 1996; Lippincott, 1995; Zona, 2002). In a large multi-year collaborative effort, Fairchild Tropical Botanic Garden (FTBG), Florida Department of Environmental Protection (FDEP), and the National Park Service (NPS) augmented the species' single wild US population and reintroduced *P. sargentii* to several historic locations (Lippincott, 1995). All sites are now federal and state public parks where threatened flora and fauna are protected. The reintroduction of *P. sargentii* to federal and state parks required careful documentation of the species' historic distribution and was a landmark, because it was the first plant reintroduction to occur in a Florida state park.

To determine whether the reintroductions have successfully established self-sustaining populations, we examined growth and dynamics of the wild and reintroduced populations of *P. sargentii* in a trend analysis (e.g., Pavlik, 1994). We specifically asked the following questions: How does survival of plants in the wild population compare to that of reintroduced populations? What factors influenced plant growth and survival of reintroduced plants, i.e., transplant location (site and microsite), year, or plant size at time of transplant? Are the historical locations (each of the three keys) appropriate for sustaining the populations now? How do the reintroductions influence overall stage structure and population viability of the species?

2. Material and methods

2.1. Species and study area

P. sargentii is a slow-growing long-lived perennial palm native to the Caribbean Basin. Seedlings have lanceolate entire leaves called eophylls (Tomlinson, 1960). With maturation, adults have single erect, cylindrical stems that can achieve heights of 8 m and have alternate, spirally arranged pinnately compound fronds (Zona, 2002). Although *P. sargentii* has reached reproductive maturity in cultivation within 14 years (R. Campbell, personal observation), wild individuals that are not supplemented with water or fertilizer may take much longer to reproduce. Produced in December through February, fruits are red spheroids with oily endosperm when ripe in January through April (Garvue and Carrara, 2001), and are likely to be animal-dispersed (Zona, 1997). Seeds may require 6–24 months to germinate after removal or cracking of endocarp (Garvue and Carrara, 2001), but generally tropical palms are not believed to have a long-lived seed bank (Zona, personal communication).

Throughout its distribution, *P. sargentii* grows predominantly in coastal habitats on limestone or dune sand over limestone in seasonally dry forest, tropical hammock, and coastal scrub (Seifriz, 1943; Ledin et al., 1959; Read, 1968; Quero, 1981). The single natural wild US population grows within Biscayne National Park (BNP) on Elliott Key, where it grows in a rockland hammock on limestone substrate (Florida Natural Areas Inventory, 2002). Historically, *P. sargentii* was known from three islands in the Florida Keys (Ledin et al., 1959) that had small populations of a few to 200 plants even at the time of its discovery in 1886 (Sargent, 1886, 1888). By 1925, it was believed the species was extirpated from the Florida Keys, but in 1950, 28 adult palms were rediscovered on Elliott Key, and eight years later three adult plants between 2-m and 4-m tall were discovered on Long Key (Ledin et al., 1959). The authors noted that they saw no seedlings at either Key. A 1991 survey of all historic US locations revealed that the palm was extirpated from Sands and Long Key, but 47 plants remained on Elliott Key (Lippincott, 1995). The species was listed as Florida endangered (Coile, 2000). Because Caribbean populations also have been extirpated or are declining (Zona, 2002), the species' IUCN status is considered to be regionally endangered and globally vulnerable (Johnson, 1996).

To secure the US population, FDEP and FTBG scientists conducted reintroductions of the species from nursery grown individuals in 1991–1993 (Lippincott, 1995). Into 13 locations within its current and historic US range on BNP and FDEP lands, we introduced plants propagated at FTBG from seeds originally collected from Elliott Key (Table 1). Reintroduction sites varied in microhabitat: Elliott Key sites were all rockland hammock, Long Key sites were either coastal berm or rockland hammock and Sands Key was coastal berm (Florida Natural Areas Inventory, 2002, Table 1). Although there were no descriptions of specific microsites used historically by plants at Elliott or Sands Keys, palm stumps revealed potential historic locations of *P. sargentii* on Long Key. In addition, *P. sargentii* populations in the Yucatan Peninsula grow in all microhabitats selected for our reintroductions, ranging from sandy substrates exposed to wind and salt spray, behind berms on sandy soils, to rocky substrates and in deeper soils with some accumulation of organic matter (Duran, 1995).

2.2. Reintroduction, Monitoring, and Analysis

Planting occurred from May through July, when daily rains fell reliably. In rockland hammocks, planting required picks, shovels, and steel bars to dig holes. A high organic nursery soil mixed equally with native soil and leaves back-filled holes for the largest transplants that were in 3 or 10-gallon containers. Sandy berms allowed much easier transplanting. All plants received water after transplanting, but then received only ambient precipitation thereafter. After observing heavy herbivory on plants <150 cm tall on Elliott and Long Key, we caged plants reintroduced to Long Key that were younger than four years old. Over time, we removed cages on any individual that achieved a height of 150 cm. After 1991, we did not plant on the landward side of berms, because of high mortality in that microsite.

Prior to and following introductions, FDEP and FTBG scientists monitored the wild and introduced *P. sargentii* populations. Initially hand-drawn maps guided us to reintroduced

Table 1 – Numbers of *P. sargentii* plants outplanted from 1991 through 1994 and surviving to 2004 at thirteen locations

Location	Habitat type	Year planted	No. Planted	No. Surviving	Percent survival
Long Key 1A	Rockland Hammock	1991, 1994	54	35	64.81
Long Key 1B	Rockland Hammock	1991, 1994	25	10	40
Long Key 2B	Top of coastal berm	1991, 1993	30	4	13.33
Long Key 2C	Top of coastal berm	1993, 1994	72	52	72.22
Long Key 3A	Landward side of coastal berm	1991	8	0	0
Long Key 3B	Landward side of coastal berm	1991	10	0	0
Sand's Key	Landward side of coastal berm	1991	3	0	0
Elliott Key 16	Rockland Hammock	1993	26	5	19.23
Elliott Key 17	Rockland Hammock	1993	26	3	11.54
Elliott Key 11	Rockland Hammock	1991	3	2	66.67
Elliott Key 12	Rockland Hammock	1991	4	2	50
Elliott Key 14	Rockland Hammock	1992	3	2	66.67
Elliott Key 15	Rockland Hammock	1991	1	0	0
Total			265	115	43.39

Note that reintroductions failed in some sites.

plants. By 2000, we tagged and recorded GPS coordinates on each reintroduced and wild plant. We measured height to tallest rachis, diameter at breast or crown height, number of leaves and presence/absence of growth spike of all individuals, noting survival, health, and presence of herbivory, chlorosis and/or overgrowth by surrounding vegetation. Due to the logistical difficulty of accessing plants and the 320 h required to monitor the Elliott Key population, in some years only a subset of individuals were visited and measured, however, Long Key reintroduced populations have been monitored yearly. All handwritten and electronic records for endangered species are maintained in perpetuity at our institutions. Living *P. sargentii* plants and their associated accession records are maintained at FTBG as part of the national collection of the Center for Plant Conservation.

2.3. *Data analysis*

To determine growth patterns of wild individuals, we examined leaf development and plant height at Elliott Key. We used regression analysis to determine the relationship of plant height to leaf ontogeny (SYSTAT 10.2, 2002) and categorized plants into five non-overlapping stages.

To understand the population structure and survivorship of plants in various stages of development, we assessed survival of wild and reintroduced individuals in each stage.

For reintroduced plants, we calculated overall survival from time of transplanting to 2004 for plants in different stages, sites, and years using Kaplan–Meier survival analysis (SYSTAT 10.2, 2002). We examined growth of plants of different stages at the reintroduction sites on Long Key using analysis of variance, where transplant year and site were the main fixed effects and stage was the covariate (SYSTAT 10.2, 2002).

To understand how the reintroductions influenced overall stage structure and population viability of the species, we developed a five-stage model and assessed survival of wild and reintroduced individuals for the four-year transition period 2000–2004. We developed two transition matrices. The first used percent survival, transitions to larger stages, and reproductive values for wild plants in five stages (Table 2). To examine how the reintroduced plants influenced the species' population trajectories, the second matrix used combined percent survival, growth transitions, and reproductive values of wild and all reintroduced plants in five stages from 2000 to 2004 (Table 3). Because not all five stages are represented in the reintroduced population and no reproduction has

Table 2 – Stage-based matrix for survival and transition frequencies from 2000 to 2004 of five stages of *P. sargentii* in the wild population

	Stage 2000				
	Seedling	Sm. Juvenile	Med. Juvenile	Lg. Juvenile	Adult
Stage 2004					
Seedling	0.25	0	0	0	11
Sm. Juvenile	0.55	0.887	0	0	0
Med. Juvenile	0	0.037	0.704	0	0
Lg. Juvenile	0	0	0.295	0.99	0
Adult	0	0	0	0.001	0.917

Stages are defined as follows: Stage 1 = seedlings: seedlings with no pinnate leaves and only eophylls present; Stage 2 = small juveniles: plants with pinnate leaves and eophylls present with height $\leqslant$ 0.83 m; Stage 3 = medium juveniles: plants >0.83 m and $\leqslant$ 1.6 m with no eophylls and no flowers present; Stage 4 = large juveniles: plants 1.6 < 6.5 m and basal diameter $\geqslant$ 0.9 cm with no eophylls and no flowers; and Stage 5 = adults: plants with woody base and flowers present. Initial vector was [20 133 44 12 12].

Table 3 – Stage-based matrix for transition frequencies from 2000 to 2004 of five stages of *P. sargentii* in the combined wild and reintroduced population

	Stage 2000				
	Seedling	Sm. Juvenile	Med. Juvenile	Lg. Juvenile	Adult
Stage 2004					
Seedling	0.292	0	0	0	11
Sm. Juvenile	0.542	0.714	0	0	0
Med. Juvenile	0	0.203	0.688	0	0
Lg. Juvenile	0	0.016	0.3	0.99	0
Adult	0	0	0	0.001	0.917

Stages are defined as in Table 2. Initial vector was [23 174 77 34 12].

occurred, it was not possible to generate a separate matrix for the reintroduced population alone. Because only 14 reintroduced plants have survived on Elliott Key, we pooled all reintroduced plants on both keys with wild plants to provide an adequate sample size for the combined model. We assumed that all seeds germinated within the four-year interval as we had no data indicating that seeds will germinate after 24 months. For large juveniles, neither mortality nor transitions were seen in the past four years in the wild population or the combined wild and reintroduced population, therefore, we decreased the stasis element to 0.99 and added the conservative value of 0.001 to the growth cell. As mentioned earlier, transition from large juvenile to adult may require >14 years to achieve. Using the stochastic simulation program RAMAS GIS (Akcakaya, 2005) to generate models with 1000 simulations over 120 years, we calculated population growth, trajectories of each stage, extinction risk, elasticities, and sensitivities. Model assumptions included scramble competition-type density dependence that would affect all vital rates; standard deviations were 10% of all vital rates; and there was a 0.1 probability that catastrophic hurricanes would affect abundance of all stages. Because we have only a single transition, we do not have an empirical measure of variation, therefore we chose 0.1 standard deviation as a conservative estimate of low variation. Although hurricanes are frequent in our region, between 1991 and 2006, there have been two hurricanes that directly affected Long Key populations and a single hurricane that directly affected Elliott Key or an average of 0.1. Using the 2000–2004 interval probably generated an optimistic model (e.g., Schwartz, 2003), but this interval represented the two years when we had full surveys and measurements of the wild population and all living reintroduced plants.

3. Results

As *P. sargentii* grows, its leaf ontogeny changes, therefore we classified individuals into stages based upon plant size and leaf morphology. Using only wild individuals for analysis, we found a negative correlation between eophyll presence and plant height ($R^2 = -0.20$, $p = 0.0001$; Fig. 1) and a positive relationship between the number of pinnate fronds and plant height ($R^2 = 0.55$, $p = 0.0001$; Fig. 1). Based upon height and leaf growth parameters, we classified *P. sargentii* into five non-overlapping stages: Stage 1 or seedlings: seedlings with no pinnate fronds and only eophylls present; stage 2 or small juveniles: plants with pinnate fronds and eophylls present

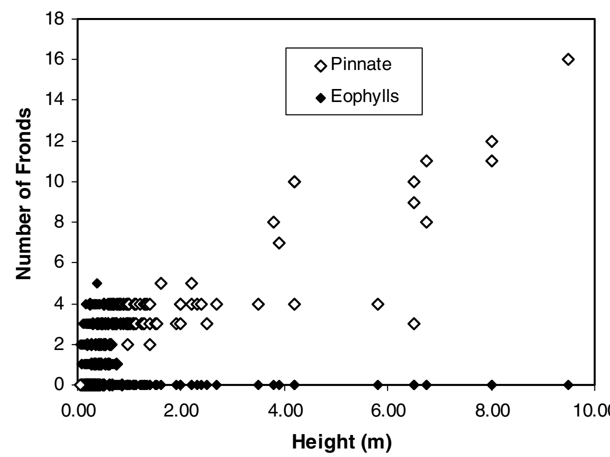

Fig. 1 – Pattern of *P. sargentii* height versus eophyll and pinnate frond production. Eophyll production of seedlings persists until plants are 0.83 m. Plants begin producing pinnate fronds after they grow to 0.19 m. At 1.6 m plants cease producing eophylls.

with height ⩽ 0.83 m; stage 3 or medium juveniles: plants >0.83 m and ⩽ 1.6 m with no eophylls and no flowers present; stage 4 or large juveniles: plants >1.6 < 6.5 m and basal diameter ⩾ 0.9 cm with no eophylls and no flowers; and stage 5 or adults: plants with woody base and flowers present. Adult plants producing flowers had heights ranging between 3.8 m and 9.5 m and had an average of 10 fronds.

In 2000, we observed 221 plants in the wild. By January 2004, the wild population at Elliott Key totaled 302 plants, most of which were small juveniles (Fig. 2). Sixty-five plants were new to the census in 2004 either due to germination or first detection by researchers. The wild population has had good survival of all plants from 2000–2004 (93%) and has increased over 6.4-fold since the 1991 survey by Lippincott (1995). Adults, medium and large juveniles experienced low levels of mortality, while the smallest size classes had 6–19% mortality (Table 2). A single adult died from damage by a fallen poisonwood tree (*Metopium toxiferum*). After 12–14 years, reintroduced plants have increased the total plants in the wild by 27% and have contributed significantly to the medium and large juvenile stages (Fig. 2).

Since transplanting, reintroduced plants have had variable survival depending upon reintroduction site, stage, and year of transplant. The overall survival of transplants was 43%,

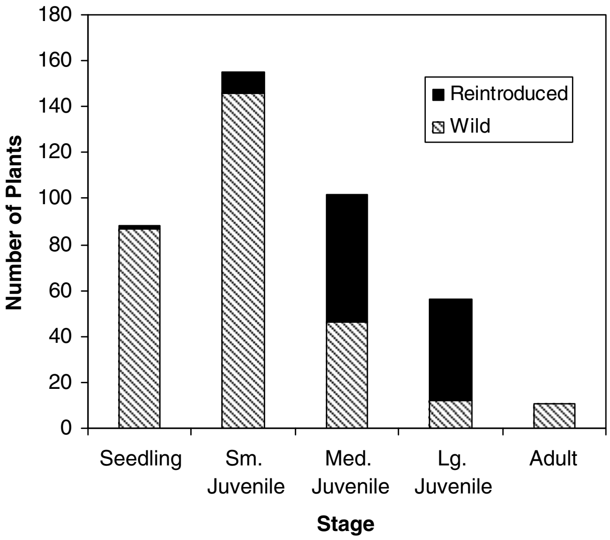

Fig. 2 – Total wild and reintroduced *P.sargentii* plants in five stage classes in 2004. The wild population has 302 plants and reintroduced populations have 110 plants. The wild population occurs on Elliott Key and reintroduced populations are on Elliott and Long Keys.

but ranged from 0% to 72% (Table 1). Sites had significantly different growth curves (Kaplan–Meier $\chi^2 = 91.23$, $p = 0.0001$). All landward side of coastal berm sites failed to support plants, but plants planted at the top of coast berms and seven of eight rockland hammock sites had survival (Table 1). The microsite with the greatest survival was the top of the coastal berm on Long Key.

Initial stage of plants also significantly influenced survival of transplants (Kaplan–Meier $\chi^2 = 38.97$, $p = 0.0001$; Fig. 3). The largest individuals had the greatest survival and the smallest individuals had the lowest survival. However, the relationship between plant size and survival was not linear, as small juveniles had greater survival than medium juveniles after 14 years. The majority of medium juveniles were transplanted into sites 16 and 17 on Elliott Key, where herbivory was high confounding our ability to assess the direct affects of plant size on transplant survival. On Long Key plants <150 cm tall were caged to prevent herbivory. No seeds introduced to Long Key 2b germinated and they were not included in the survival analysis. Note that currently no reintroduced plants are reproductive adults (Fig. 2).

Time of transplant significantly influenced survival. Transplants in 1991 had significantly lower survival than those planted in 1992, 1993 or 1994 (Kaplan–Meier $\chi^2 = 32.78$, $p = 0.0001$, Table 1). Plants transplanted in 1992 and 1994 had the greatest survival.

Growth of transplants varied with reintroduction site. For the Long Key reintroductions, where we had yearly growth measurements, transplant site and year of transplant significantly influenced plant growth (Site $F_{3106} = 8.11$, $p = 0.0001$; Year $F_{2106} = 6.36$, $p = 0.003$; Fig. 4). Initial stage was not a significant covariate ($F_{1106} = 3.69$, $p = 0.058$) and there was no significant interaction ($F_{1106} = 0.559$, $p = 0.45$). The greatest growth occurred in plants growing on the top of coastal berms at site 2c and the least occurred in a rockland hammock at sites 1a and 1b. The tallest plant on Long Key was 286 cm in 2004; it was 58 cm in 1993 when it was planted at site 2C. However, there was not a clear relationship between growth rate and transplant survival at a site (Table 1, Fig. 4).

Growth rate of *P. sargentii* is slow, as are transitions between stages, but a greater proportion of reintroduced plants transition to larger stage classes than in the wild population alone (Tables 2 and 3). In 2004, two plants 204 cm and 238 cm tall began developing wood; they were 109 cm and 158.5 cm when planted at 1b and 1a in 1991. Given an average growth of 10 cm/yr (Fig. 4) and reproductive maturation at 3.8 m, which is the shortest reproductive individual in the wild population, plants on Long Key will require 14–17 more

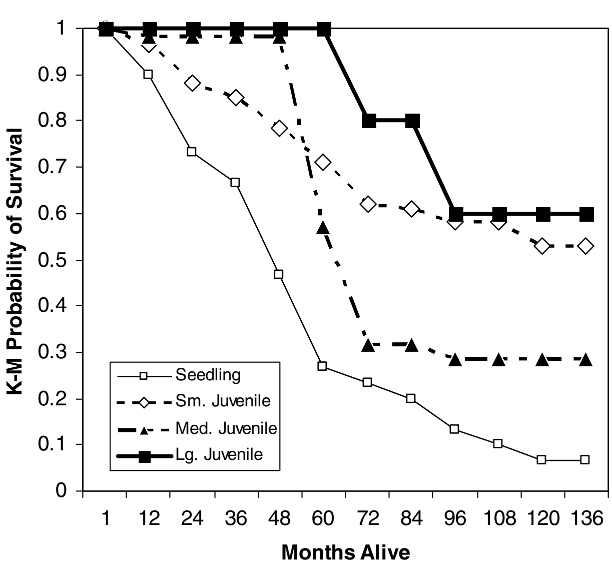

Fig. 3 – Survival curves of plants in four stages at time of transplant reintroduced to Long and Elliott Keys. Note there are no adult plants in the reintroduced populations.

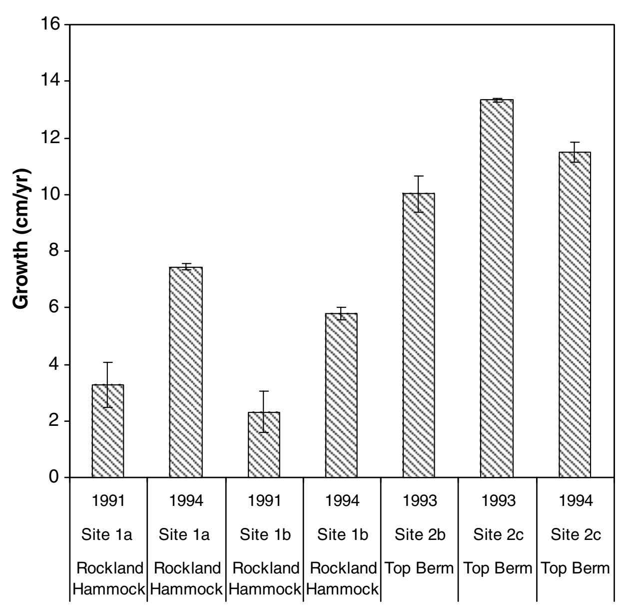

Fig. 4 – Average growth per year for plants reintroduced to 4 locations in 3 years on Long Key. Year of introduction, location, and habitat types are indicated.

years to reach reproductive maturity. Some reintroduced small juveniles, however, have become large juveniles within 4 years. This rapid growth has not been seen in the wild population and influenced the PVA for the combined wild and reintroduced model.

PVA models affirm that the wild population is growing ($\lambda = 1.013$; Fig. 5). Elasticity values indicated that the large juvenile stasis element contributed most to the population growth rate (0.6411). Sensitivity analysis indicated that transition to reproductive stage would have great influence on population growth, although it should be noted that we have not observed any plants transition to reproductive stage in 14 years. There was no risk of extinction predicted within the next 120 years.

When reintroduced plants were added to the model, vital rates changed and a greater percentage of plants made transition to larger stage classes (Table 3). The population had improved viability with a greater lambda value ($\lambda = 1.032$), yet higher variation especially after 40 years (Fig. 6). Reintroduced plants experienced more rapid growth and improved the proportion of larger individuals in the population such that the number of large juveniles after 120 years was 3-fold greater in the combined model than the wild only model. Elasticity and sensitivity analyses reflected the same findings as for the wild only model. Note that the variation for the combined model completely encompassed the wild only model within the first 70 years. As was true with the wild only model, there is no risk of extinction predicted within the next 120 years.

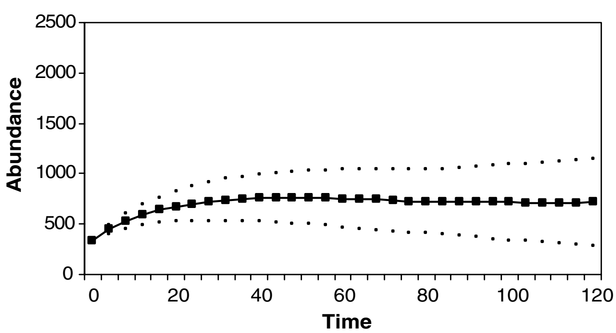

Fig. 5 – Projected *P. sargentii* wild population size based on population viability models generated with 1000 simulations run for 120 years under scenarios of scramble competition, density dependence affecting all vital rates, 10% standard deviations of all vital rates, and 0.1 probability that catastrophic hurricanes would affect abundance of all stages. Average value (bold squares) and ±1 standard deviation are displayed.

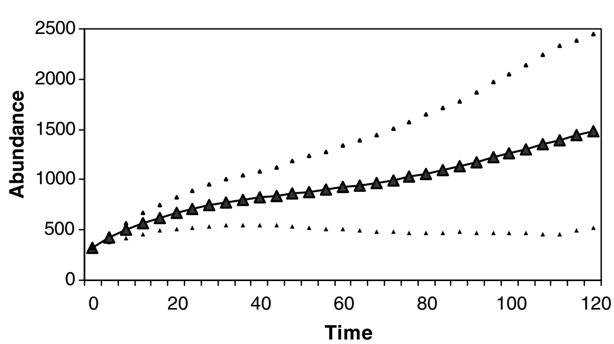

Fig. 6 – Projected wild and reintroduced *P. sargentii* population size based on population viability models generated with 1000 simulations run for 120 years under scenarios of scramble competition, density dependence affecting all vital rates, 10% standard deviations of all vital rates, and 0.1 probability that catastrophic hurricanes would affect abundance of all stages. Average value (bold triangles) and ±1 standard deviation are displayed.

4. Discussion

The US *P. sargentii* wild population has had positive growth and is recovering well. Since 1991 when only 47 plants existed in the US, there has been a 6.4-fold increase in total numbers. This is due to good seedling recruitment and the elimination of harvesting and habitat destruction. Although the species was formerly threatened by over-collecting, now its availability in nurseries and remote, difficult accessibility to wild and reintroduced populations have minimized this threat. We have not observed any collections of wild or reintroduced individuals during our 14 year study. In addition, the wild populations are now growing in public parks that are protected from development. Population viability analysis indicates the population has no risk of extinction.

The wild population has a skewed age structure like many long-lived species (e.g., Schwartz, 2003). Most of the population is relatively small in stature and young in age. Very few adults hold the reproductive capacity of the population. Because plants may take decades to reach reproductive maturity, adult death greatly impacts the reproductive capacity of the population. Lippincott (1995) observed that the tallest individuals were most susceptible to dying from hurricane force winds. In 1992, shortly after Hurricane Andrew passed directly over the wild and reintroduction populations on Elliott Key with wind speeds exceeding 140 mph (125 knots), it killed 19 of the tallest palms in the wild population and two of the reintroduced palms, which were much smaller in stature. Hurricanes can also impact the smallest stages directly from tidal surge and indirectly from deep litter accumulation from leaf drop after salt deposition on hammock plants (e.g., Tanner et al., 1991; Pascarella, 1998). Reintroductions conducted in 1991 had significantly lower survival, and it is likely that Hurricane Andrew decreased long-term survival of smaller stature plants due to these factors. Because hurricanes are major and frequent natural disturbance in the Florida Keys, storms make the small *P. sargentii* population vulnerable to losing its reproductive adults and/or having greater mortality of the smallest individuals.

The reintroductions have greatly improved the mid-range age structure and viability of the US population, and have expanded the species' distribution to historical sites. At least some reintroduced individuals are exhibiting faster growth than those in the wild population, which contributes to the PVA model predictions that the population with reintroductions is growing at a faster rate than the wild population alone. Reintroductions improved the representation of larger individuals to population growth. For example, the predicted

3-fold increase in large juveniles within 60 years improves the probability that at least some individuals with reproductive capacity will be alive within the next 100 years even with hurricane disturbance. This is especially important because maturation to reproductive adult stage is extremely slow. A 19-year-old plant introduced to Long Key has only just begun to produce a woody trunk and our calculations suggest that >14–17 more years may pass before the first reintroduced plants flower. Provided mortality and births remain similar to what our observations have been in the past 14 years, *P. sargentii* has no risk of extinction in the US.

As we noted earlier, using a four-year interval probably generated an optimistic model as did including a transition element for growth to reproductive adult. In a comparison of model accuracy versus duration of study for the long-lived species *Arisaema triphylllum*, Schwartz (2003) noted that 10–20 years of sampling would be required to detect whether the population was declining with 95% accuracy. Due to the longevity of *P. sargentii*, >20 years will be needed to detect transitions between stages and institutional dedication to long-term monitoring will be required to assess whether the populations are secure.

Several general lessons arise from this reintroduction. As has been seen in other experimental reintroductions and models (e.g., Guerrant, 1996), using larger plants generally improved the success of reintroductions. By using small to large juveniles in our reintroductions, our theoretical models indicated that we improved the probability of achieving reproductive maturity within the next two decades and helped improve the likelihood of persistence in the Florida Keys. Although larger plants required more institutional cost and care prior to and during reintroduction, the long-term conservation value is great. In contrast, seeds did not prove to be good reintroduction propagules. Because germination trials in the FTBG nursery conducted after the reintroduction indicated that fresh seeds may require up to 24 months to germinate (Garvue and Carrara, 2001; Garvue and Duquesnel, 2001), the lack of germination in our reintroduction trials was not surprising.

Our study emphasizes the importance of approaching restorations experimentally to help identify microsite needs of the species (e.g., Fiedler and Laven, 1996; Maschinski et al., 2004) and across multiple sites within the current and historic range of the species to buffer against stochastic losses (Maschinski, 2006). Using such an approach particularly in a region prone to major disturbance will ensure that entire populations will not be destroyed simultaneously (McEachern et al., 1994). Metapopulation theory predicts that some colonization events will succeed while others will fail (Hanski, 1999), and this should guide restoration planning to expand to multiple sites (Maschinski, 2006). Not all of our *P. sargentii* reintroduction sites had successful establishment or optimal growth for the species. This indicates either that some historic locations may no longer be suitable for supporting populations of *P. sargentii* or that some microsites are not appropriate for *P. sargentii*. At three sites (Sands Key and Long Key 3a and 3b, Table 1), no plants survived. This may be explained by the major change in topography at Long Key 3 in the past two decades; there is no longer a foredune at the site. We suspect that the plants reintroduced there were inundated with salt water after tidal surges from hurricanes. In contrast, the microsite at the top of coastal berms had the fastest plant growth rates and good survival at one location. This is proving to be very important for long-term viability of the population. Periodic introductions of nursery-grown individuals would contribute to population stability and further buffer the species from extinction.

This restoration also had a social–cultural impact and was a successful collaboration among federal, state, and non-government entities. Because these institutions have commitments of endangered species' conservation and maintain written and electronic records, collaborations on this project persisted despite staff turnover. Expertise and ex situ collection of the species at the local botanical garden were essential for the reintroduction, as was cooperation and support of the land managing agencies (e.g., Maunder et al., 2001). These first rare plant reintroductions to Florida State Parks opened avenues for more plant conservation efforts. Since the successful *P. sargentii* reintroductions, more rare plant reintroductions and augmentations have been initiated in Florida state parks. The research has been interpreted to the public at the parks and botanical garden to raise awareness of and support for rare plant conservation.

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